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91-04561/

**LAMBTON FGD PROJECT: ENVIRONMENTAL
IMPACT OF THE PROPOSED
GYPSUM STORAGE AREA**

Report No. 90-275-K

H.M. Johnston, Hydrogeologist
and
K.E. Eagleson, Hydrogeology Technologist
Rock Sciences Section
Civil Research Department

ABSTRACT

The implementation of the wet limestone slurry process with forced oxidation as a means of Flue Gas Desulphurization (FGD) at the Lambton TGS will result in the production of a saleable gypsum by-product. The construction of landfill facilities for by-product storage requires an assessment of the potential impact of the proposed gypsum storage area on the local groundwater environment. This report discusses the predicted chemical characteristics of the FGD material, identifies potential contaminants and addresses the potential environmental impact of these solutes. Estimated penetration depths for boron, selenium, arsenic and cadmium calculated using effective diffusion coefficients indicate the clay till is a substantial barrier to the downward migration of potential contaminants.

job	44-242-473	file	841.11	date	November 9, 1990	report no.	90-275-K
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To Dr. W.R. Effer
Manager
Environmental Studies Department
H10G03

Attention: D.E. Barker, H10E05

**LAMBTON FGD PROJECT: ENVIRONMENTAL IMPACT OF THE
PROPOSED GYPSUM STORAGE AREA**

1.0 INTRODUCTION

Flue Gas Desulphurization (FGD) equipment will be installed at the Lambton Thermal Generating Station (TGS) in order to reduce sulphur dioxide emissions from the station. The first set of scrubbers (Units 3 and 4) are scheduled to go into service in early 1994 with the remaining two units having a 1996 in-service date. The FGD process chosen for use at the Lambton TGS is the wet limestone slurry process with forced oxidation. This process allows for the production of a saleable gypsum by-product suitable for the manufacturing of wallboard.

Gypsum in excess of contractual requirements as well as that which does not meet the specifications of the manufacturers will be placed in two proposed landfill sites located in the immediate vicinity of the station. Impacts to the local surface and groundwater environments may occur by the off-site movement of surface run-off and/or leachate from the gypsum landfill. The purpose of this report is to briefly describe the chemical characteristics of the solid gypsum and its leachate and to assess the potential environmental impacts.

2.0 FGD BY-PRODUCT CHEMICAL CHARACTERISTICS

2.1 Solid FGD Material

The chemical properties of the waste form are highly dependent on the scrubbing agent, the composition of the coal and the percentage of fly ash entrapped (Ainsworth and Rai, 1987). However, the solid material produced by limestone wet scrubbing with forced oxidation is expected to consist primarily of calcium sulphate (gypsum: $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) and unreacted limestone reagent (calcium carbonate: CaCO_3) with minor amounts of calcium sulphite (hannebachite: $\text{CaSO}_3 \cdot \frac{1}{2}\text{H}_2\text{O}$) and fly ash. Approximately 94% to 96% of the total solid phase is predicted to be in the form of gypsum with between 1% and 2% unreacted limestone, a similar percentage of CaCO_3 (formed during the scrubbing process) and between 0.6% and 3% fly ash (Golomb, 1990). Estimates of the minor element content of the solid FGD material is given in Table 1. The calculated ranges are based on the anticipated levels of fly ash entrapment (0.6-3%) and the average elemental analyses of Ontario Hydro post-precipitator ash reported by Curtis (1977). As seen from Table 1, the estimated elemental composition of Lambton FGD material is within the lower limit of the range of elemental concentrations reported for U.S. FGD sludges (Rai et al., 1987).

2.2 Initial FGD Leachate

Although the chemical characteristics of the FGD material will be dependent on several factors and is therefore site specific, with respect to major ion chemistries, gypsum produced as a by-product from the FGD process is anticipated to be generally similar to natural gypsum. Gypsum solubility is fairly high ($K_{eq} = 10^{-4.5}$), therefore leachate generated from gypsum storage areas will be gypsum saturated having elevated total dissolved solids (TDS) content with Ca^{+2} and SO_4^{-2} as the dominant ions.

The rates and amounts of the minor elements released to solution due to leaching are a function of a number of factors including the initial concentration of the element in the material, the distribution and form of the element in the solid phase and the chemical nature of the leaching solution. In order to provide a qualitative estimate of the most readily leachable components in the FGD material, the estimated elemental composition of minor elements given in Table 1 were coupled with the maximum water-extractable data for fly ash presented by Eary et al. (1990) along with the assumption that only 1% of the leachable elements in the fly ash remain in the aqueous phase after the wash procedure (99% efficiency in rinsing of FGD material prior to landfilling; Golomb, 1990). Predicted

aqueous concentrations of minor elements in the initial porewaters associated with the FGD material are given in Table 2; Provincial Drinking Water Objectives and Guidelines (PDWO/G) are also listed. Note that, except for chloride and selenium, the lower end of the range of the predicted concentrations in the FGD leachate are below the PDWO/G. If the upper end of the concentrations is considered, arsenic (As), boron (B), cadmium (Cd), chloride (Cl), chromium (Cr), manganese (Mn) and selenium (Se) exceed the drinking water standards. In addition to these solutes, sulphate and TDS concentrations in the leachate will be well above the specified guidelines for drinking water quality. It is significant to note that the upper end of the concentrations given in Table 2 is well below 100 times the maximum allowable level for parameters related to health effects and, therefore, would likely be classified as a solid non-hazardous waste material under MOE Regulation 309.

The concentration levels given in Table 2 are estimates of the initial interstitial water associated with the FGD material at the time of disposal/storage. As such it is anticipated that these levels are conservative i.e. the concentration of the majority of the trace elements, as well as Cl, will decrease due to the dilution effects of infiltrating rainwater and groundwater. For example, Knight et al. (1980), in a study of unfixed FGD sludge, reported that the TDS content of the leachate decreased to a level controlled primarily by the solubility of gypsum after being eluted with about five pore volumes of water. The potential impact of these contaminants on the local groundwater regime at the proposed gypsum disposal site is discussed in Section 3.4.

2.3 Quality of Surface Water Run-off

During the operation of the gypsum landfill when the FGD by-product is actively being placed, the FGD material will be exposed to precipitation which in turn will lead to run-off. This run-off must be contained and, depending on the water quality, treated before it is released.

Included in Table 2 is a listing of the MOE water quality objectives for surface water bodies. A comparison of the concentration limits presented in column 3 with the initial FGD liquid phase concentrations indicate several species are likely to be present in excess of the given objectives and thus the water would require treatment prior to release. However, it should be noted that significant dilution of the initial FGD leachate will occur due to mixing with precipitation. For example, water analyses were conducted on run-off

samples taken from a gypsum disposal site at the Tennessee Valley Authority's Shawnee steam plant in Kentucky by Hurt et al. (1981). They reported the initial run-off water to contain 2500 mg/L total dissolved solids (TDS) of which the bulk was comprised of SO_4^{2-} (1500 mg/L), Ca^{2+} (~600 mg/L) and Cl^- (<500 mg/L). The dilution potential of precipitation is evident when the initial FGD porewater concentrations are considered. For example, the TDS content in the sludge liquor was reported to be ~10,000mg/L and chloride levels were on the order of 3500 mg/L, representing an initial dilution factor of approximately 4. In addition, the concentrations in the run-off continued to decrease over time, particularly in the case of Cl which, after a period of one year, decreased to below 50 mg/L giving an overall dilution factor of ~70. The TDS levels were also observed to decrease, however, not as dramatically (down to ~2200 mg/L) due to the control of the run-off chemistry by gypsum solubility.

From the results presented by Hurt et al. (1981), it is anticipated that once the initial flush of leachate in the surface zone of the gypsum has been removed by precipitation, subsequent water quality will be dominated by calcium and sulphate ions and may not require treatment. However, site specific data, i.e. collection and analyses of surface run-off, will be needed to determine whether treatment will be required prior to discharge of the run-off water to local surface water bodies.

3.0 HYDROGEOLOGIC CONDITIONS AT THE PROPOSED DISPOSAL AREA

3.1 Stratigraphy of the St. Clair Clay Plain

The proposed FGD storage area is situated in the St. Clair clay plain which covers all of Lambton County, extending eastward from the bank of the St. Clair River a distance of 50 km, and from Sarnia southward 30 km and southwestward 60 km. With respect to topography, subtle irregularities, termed ridges and swales by Gartner Lee Limited (1987), appear on the relatively flat St. Clair clay plain. These features, common to clay plains in other parts of North America (Keller et al. 1986), have a relief of less than one meter and a spacing in the order of tens of meters. From air photographs taken of the Tricil site (Richards et al., 1984), it is apparent that a sub-surface lateral flow system is present which is influenced by the small local topographic features resulting in the creation of wet and dry areas. Regionally, the land surface slopes gently westward toward the St. Clair River and the underlying sedimentary bedrock also dips westward.

The clayey sediment of the overburden is of Quaternary origin and ranges in thickness from 25 to 50 metres. Comprised of two units of till known as the St. Joseph till and the black shale till, the overburden can generally be divided into two zones (a weathered crust and an unweathered massive zone). The upper till unit (St. Joseph till) is 7 to 13 metres thick and contains significant amounts of clay (30 - 60%) and silt (30 - 40%) with lesser quantities of sand (5 - 10%) and gravel (<5%) (Hanna, 1966; Harding, 1986; McKay and Trudell, 1989). The lower unit known as the black shale till is a silty clay till which ranges in thickness from 30 to 40 metres and contains abundant black shale clasts. The source of these clasts is likely from the underlying bedrock - a thinly bedded, westward dipping, dark bituminous shale of Devonian age known as the Kettle Point Formation. A stratigraphic cross-section of the overburden and bedrock is shown in Figure 1.

Weathering effects extend 4 to 6 metres below the surface within the St. Joseph till unit. The weathered zone can be most easily identified by its buff brown colour, likely due to the oxidation of ferrous iron in chlorite (D'Astous et al., 1989). The alteration of illite and chlorite to smectite is also evident. The mineralogical composition of the weathered clay till is mainly illite (~50%), with smaller amounts of smectite (15%), quartz and feldspar (20%), carbonates (10%) and chlorite (5-8%) (UWO, 1990). Due to the weathering effects, this zone has become brittle and is intensely fractured in the upper 4 metres and less fissured from 4 to 6.5 metres (UWO, 1990). Fractures have been observed to depths of 7.7 metres (Abbott, 1987) and roots at depths of up to 9 metres (Hanna, 1966). Based on the analysis of tritium profiles and water-level fluctuations, Ruland et al. (1990) determined that the maximum depth of hydraulically-active fractures extends to depths as much as 10 m below ground surface which is 4 to 5 m below the weathered zone. These fractures are often found filled with medium to coarse grained secondary gypsum crystals which are indicative of fluid flow. Detailed observations of the fracture patterns were made on the walls of two trenches which were oriented at right angles to each other. Two sets of fractures were identified in the weathered zone (Desaulniers, 1984). One set of fractures has a mean azimuth of 75° while the other set is oriented at approximately right angles to the first set with an azimuth of 140°. The spacing of the fractures ranges between 3 to 10 cm while the fracture apertures were generally less than 0.5 mm. Results of a tracer experiment performed at the Tricil site by D'Astous et al. (1989) provided estimates of fracture apertures of between 0.02 and 0.03 mm.

Dark, clay-rich glacial till of the unweathered zone extends below the weathered zone to a depth of approximately 50 metres, ranging in thickness from 30 to 40 metres. It is generally a massive, grey, silty clay which is slightly pebbly and contains thin lenses of

glaciolacustrine sandy silt material. These lenses of sandy material are isolated, hydraulically unconnected deposits (Fitzgerald et al., 1979). Carbonates are the predominant mineral type in the massive zone (35%) with other main components being illite (25%) and chlorite (22%). Quartz and feldspar comprise 17%, while smectite is present in only minor quantities (1%) (UWO, 1990). The natural organic carbon content of the till, as determined from two samples taken from the Tricil site, was 0.68% and 1.95% (McKay and Trudell, 1989).

3.2 Physical Hydrogeology

The vertical distribution of hydraulic head, as determined at two piezometer nests at the Lambton TGS site, reveals an overall decrease in hydraulic head with depth (Heystee et al., 1982). In general, both piezometer nests showed that in the upper 6 m of the deposit, the hydraulic head distribution is very close to static conditions, whereas downward flow conditions prevail below this zone (Desaulniers et al., 1982; Quigley and Rowe, 1986).

The mean downward hydraulic gradient at two locations in the unweathered till at the Lambton TGS site was found to be 0.24 and 0.48 (Heystee and Johnston, 1984) whereas Quigley and Rowe (1986) reported vertical gradients of 0.17 to 0.25 at a sanitary landfill location near Sarnia, and Johnson et al. (1989) measured a downward gradient of 0.28 in the unweathered till at the Tricil site. A horizontal hydraulic gradient determined at one location at a depth of 6 m at Lambton TGS was found to be 0.03 in the direction of the St. Clair River.

The hydraulic conductivity of the clayey till in the area has been determined by both field and laboratory techniques by several investigators. In the weathered till, laboratory hydraulic conductivity values on the order of 10^{-7} to 10^{-9} cm/s were measured and compare well with field values obtained from augered boreholes (Desaulniers, 1986; Harding et al., 1986). The magnitude of these K values is comparable to those determined for the clayey till of the unweathered zone, that is 10^{-7} to 10^{-9} cm/s, and are representative of the bulk hydraulic conductivity of unfractured Sarnia area clay (Goodall and Quigley, 1977; Richards et al., 1984; Desaulniers et al., 1986; Johnson et al., 1989; UWO, 1990). The piezometers used to obtain hydraulic data were standpipes that were placed in holes created by augers. In this type of installation no attempt is made to remove the smearing of clay on walls of the borehole caused by the augering process. As a result, piezometers rarely show evidence of the hydraulic connection to fractures.

To overcome this problem, D'Astous et al. (1989) performed recovery tests in a large diameter well and in shallow test pits at the Tricil site. Hydraulic conductivities in the range of 10^{-5} to 10^{-6} cm/s were calculated. These K values are two or more orders of magnitude higher than those calculated for the unweathered clay indicating that fracture flow is significant in the weathered zone.

Laboratory tests were also conducted to determine the porosity of the clay till. Desaulniers et al. (1982) reported porosities of the unweathered till at the Lambton site to be on the order of 0.4 to 0.45 and determined that the effective or transport porosity was essentially equal to the total porosity. Johnson et al. (1989) reported slightly lower porosity values ranging from 0.32 to 0.39 for unweathered till at the Tricil site.

Using the modified Darcy equation and representative values for the hydraulic conductivity, hydraulic gradient and porosity, average linear groundwater velocities can be calculated. Estimates of the vertical groundwater velocity in the unweathered clay till in the Sarnia area are on the order of 0.1 to 0.52 cm/a (Johnston et al., 1985; Quigley and Rowe, 1986; Johnson et al., 1983).

Groundwater velocities of this magnitude generally imply that the transport process is dominated by the mechanism of molecular diffusion (Gillham and Cherry, 1982) where diffusion is the process by which a solute moves from areas of high concentration to areas of low concentration. Indeed, from analysis of chloride migration rates at the Tricil site, Johnson et al. (1989) determined that advection is insignificant as a transport mechanism compared to diffusion. This conclusion was also reached by Desaulniers (1986) who found that isotope and Cl profiles measured at the Lambton site were consistent with one dimensional solute migration based on molecular diffusion. The relative importance of advective versus the diffusive transport process at low groundwater velocities is illustrated in Figure 2. This figure describes the position of the centre of mass of a solute plume (defined as $C/C_0 = 0.5$, where C/C_0 is the normalized concentration with respect to an initial input concentration of C_0) as a result of the diffusive transport mechanism compared to an average linear groundwater velocity of 1 cm/a. From Figure 2 it can be seen that for groundwater velocities on the order of 1 cm/a or less, the diffusion process has a significant influence on the migration rate of a solute. The actual migration rate for the solute is substantially larger than would be predicted on the basis of advection alone.

The evaluation of potential contaminant migration rates beneath the FGD storage/disposal area therefore requires estimates of the effective diffusion coefficients for the contaminants of interest.

4.0 POTENTIAL ENVIRONMENTAL IMPACT OF FGD LEACHATE

As mentioned in Section 2.2, the solutes of concern with respect to the potential degradation of the local groundwater quality are sulphate, chloride, boron, cadmium, chromium, manganese, arsenic and selenium. In order to assess the potential impact of these solutes on the local groundwater regime at the proposed disposal site, the chemistry of the natural groundwater must be considered. Table 3 summarizes the groundwater chemistry at the Lambton TGS site. With respect to the major anions, SO_4^{2-} and Cl^- , the groundwater chemistry is characterized by high sulphate (as high as 8310 mg/L) and fairly high chloride (up to 809 mg/L). The high Cl^- content in the till is a result of the upward diffusion of chloride from the bedrock aquifer, thus concentrations increase with depth. According to a water resources survey of the area conducted by Mellary and Kilburn (1969), water analyses on groundwater from the deep till or shallow bedrock zone indicate the high Cl^- concentrations are not uncommon in the Sarnia area. Sulphate concentrations, on the other hand, are highest within the upper 10 m of the overburden and decrease with depth. Results of isotopic analysis and mass balance calculations by Abbott (1987) have shown that the elevated SO_4^{2-} concentrations are due to the oxidation of bedrock derived pyrite in the till. Since the predicted interstitial water concentrations for Cl^- and SO_4^{2-} are approximately 950 mg/L and 2000 mg/L for the FGD material, it is unlikely that the FGD leachate will cause significant contamination of the local groundwater regime. It should also be noted that the predicted concentration levels for solutes given in Table 2 are likely upper limits. That is, for several of the parameters, including chloride, the actual concentration levels will decline significantly over time as the "first flush" of the leachate is removed from the system. Sulphate levels, however, are not expected to decline significantly over time since its concentration in the porewater will likely be controlled by gypsum solubility.

Unlike the major anions, the trace elemental concentration for B, As, Se, Cd, Cr and Mn are expected to be higher in the initial gypsum porewaters than in the groundwater. For example, boron concentrations in the initial interstitial waters are predicted to be in the range 0.7 to 6.4 mg/L (Table 2). The upper concentration in this range is slightly above provincial drinking water objectives (5 mg/L) and will decrease as "fresh" water percolates

through the FGD material. Nevertheless, boron is considered to be relatively nonreactive (i.e. retardation reactions are not significant) and therefore migration of boron from the gypsum into the underlying till is expected.

In order to estimate the ion-migration profile for boron, or for any of the trace metals, prediction of the transport velocities are required. In situations where molecular diffusion is the dominant transport process for contaminant movement in a porous medium, as it is in the Sarnia area till, the diffusion coefficient of the solute of interest can be used to characterize the transport mechanism. In this case, the more general transport equation, which accounts for both advection and dispersion, reduces to a diffusion equation. In its one dimensional form, this equation can be written:

$$\frac{\partial C}{\partial t} + \frac{\rho}{\eta} \frac{\partial q}{\partial t} = D_m \frac{\partial^2 C}{\partial x^2} \quad (1)$$

where C is the concentration of the solute in solution; D_m is the molecular diffusion coefficient in the medium; x is the Cartesian space coordinate; t is time; q is the concentration of the solute on the solid phase; η is the porosity; and ρ is the bulk density of the porous medium. This equation states that the increase in contaminant concentration within a volume of medium is equal to the increase in mass due to diffusive transport minus the mass of contaminant removed from the solution phase by chemical reactions. For example, if the relationship between q and C is linear, reversible and non-hysteretic, Equation (1) can be rewritten as:

$$\frac{\partial C}{\partial t} = D_e \frac{\partial^2 C}{\partial x^2} \quad (2)$$

where D_e is the effective diffusion coefficient of a solute in the medium and is defined by:

$$D_e = \frac{D_m}{1 + \left(\frac{\rho}{\eta}\right) K_d} = \frac{D_o \tau}{R} \quad (3)$$

In Equation (3), K_d is the distribution coefficient and is defined as the equilibrium concentration of solute in the solid phase divided by the concentration in the solution phase, and is applicable for reactions that are instantaneous and reversible and in which there is a functional relationship between the solution and solid phase concentrations. The denominator in Equation (3) is collectively known as the retardation factor (R), τ is the tortuosity factor and D_0 is the free solution diffusion coefficient.

For solutes such as sulphate, chloride and boron which are considered to be "conservative", the influence of chemical reactions is negligible, i.e. $K_d=0$ and, therefore, $R=1$. Arsenic, selenium, cadmium and manganese, however, are expected to be retarded significantly due to sorption and/or precipitation processes. This is evident from Table 4 which summarizes the effective diffusion coefficients for a number of ionic species. The conservative species such as chloride, boron and tritium (^3H), for example, have effective diffusion coefficients on the order of $10^{-6} \text{ cm}^2/\text{s}$ while species for which sorptive processes are significant, such as As, Se and Cd, are characterized by effective diffusion coefficients that are 1 to 4 orders of magnitude lower.

The implications of the magnitude of the diffusion coefficient on solute migration is illustrated in Figures 3 to 6 for B, Se, As and Cd, respectively. The curves on the figures represent the predicted normalized concentration distributions of the various solutes in a low velocity medium (such as the clay till in the Sarnia area) after 10, 25 and 100 years. If we consider a C/C_0 concentration level of 0.1, after a period of 10 years the advance of boron would be on the order of 129 cm, selenium would have migrated less than 14 cm, while the Cd and As concentration fronts would have advanced approximately 3 cm and 1.4 cm, respectively. Similarly, after 200 years, the C/C_0 front for B, Se, Cd, and As would have migrated 410 cm, 43 cm, 9.5 cm and 4.5 cm, respectively. When considering the predicted concentration profiles shown in Figures 3 to 6, it is important to consider the particular normalized concentration that would be of significance with respect to the leachate and ionic species of concern. For example, from Table 2, the upper limit of the predicted boron concentrations in the initial FGD porewaters is $\sim 6.5 \text{ mg/L}$ and the PDWO for B is 5 mg/L. This corresponds to a normalized concentration of $C/C_0=0.78$. Thus, if we consider the advancement of this concentration front, after 100 years, a boron level of 5 mg/L would have advanced only 70 cm, as shown in Figure 3. In fact, it would take roughly 200 years for the 5 mg/L concentration front of B to migrate through 1 meter of the till.

Using the same procedure for determining the normalized concentration level of environmental significance for Se, Cd and As, gives C/Co values of 0.33, 0.5 and 0.55, respectively. The approximate position of these concentration fronts, represented by the vertical lines in the respective diagrams, are 26 cm for selenium, 3.9 cm for cadmium and 1.65 cm for arsenic after 100 years. It should be noted that the concentration profiles shown in Figures 3 to 6 are based on the assumption that the source concentration (Co) of the contaminants are constant over time and the solute migration fronts shown are, therefore, conservative in nature (i.e. worst case scenarios). It is expected, as previously mentioned, that the actual concentration levels for the majority of the trace ionic species given in Table 2 will decrease over time. If this is indeed the case, then the concentration differential for a solute between the FGD leachate and the overburden porewaters will be reduced, thus decreasing the driving force for diffusive transport and reducing the depth of penetration of the various solutes.

Although no diffusion coefficients for Cr and Mn are given in Table 4, chemical reactions such as precipitation and co-precipitation, as well as sorption, will have a significant effect on the solution phase concentrations of these solutes. For example, field studies of several coal ash and FGD sludge disposal sites in the U.S.A. conducted by EPRI have shown no significant migration of Cr or Mn from the waste materials into the underlying overburden (Murarka, 1990). The PWQO for Cr of 0.05 mg/L refers to Cr(VI). The values given in Table 2 however, are for total Cr, that is, Cr (III) and Cr(VI). Harmful health effects are attributed to the hexavalent (VI) which is stable under strong oxidizing conditions. Due to the forced oxidation method employed in the scrubbing process, Cr will likely occur in the fresh FGD material primarily in the Cr^{6+} state. However, Cr(III) is the most common naturally occurring oxidation state for chromium and in this form it is not considered to be toxic. At the pH values and redox potentials of most soils, Cr(VI) is readily reduced to Cr(III) which is strongly sorbed by Al and Fe compounds (de Haan and Zwerman, 1978; Rai et al. 1988). Therefore, considering that chromium (III) is not considered to be toxic and migration would be significantly retarded due to sorption and precipitation reactions as stated by Murarka (1990), it is unlikely to be of environmental concern.

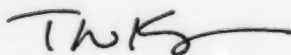
Although precipitation and sorption reactions would also be significant for manganese species, the maximum desirable concentration for Mn (0.05 mg/L) is aesthetically based. Concentrations related to health effects are much larger than those related to aesthetic considerations. The current PWQO/G concentration of 0.05 mg/L was set based on the lowest level that can be reasonably achieved through water treatment (CCREM, 1987).

In fact, although Mn concentrations in Canadian drinking water supplies are generally below 0.05 mg/L, levels as high as 4.6 mg/L have been reported (CCREM, 1987). This is ~40 times the upper concentration level predicted for the initial FGD leachate and this level will decline over time due to the ingress of both precipitation and groundwater.

The abundance of fracturing in the weathered zone at the Lambton TGS site suggests that the potential for horizontal migration may be significant. In 1982 however, a hydrogeologic study was conducted by Ash (1983) in which a network of fourteen piezometer nest, each with piezometers at depths of 3 m and 6 m, were installed along a horizontal flow path to the south of the existing coal ash disposal area. The disposal area has been in operation since 1969, yet no evidence was found to suggest that the fractures in the clay till are a significant migration pathway, at least in the short term.

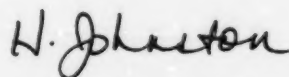
From the above investigations and discussions in addition to the fact that the overburden in the proposed disposal/storage area is approximately 45 m thick, the clayey till deposit would be a substantial barrier to the downward migration of potential contaminants.

Approved by:

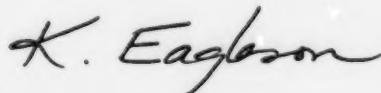


T.W. Klym
Manager
Civil Research Department

Prepared by:

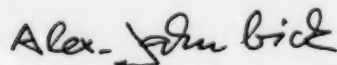


H.M. Johnston
Hydrogeologist
Rock Sciences Section
Civil Research Department



K.E. Eagleson
Hydrogeology Technologist
Rock Sciences Section
Civil Research Department

Reviewed by:



A.T. Jakubick
Supervising Engineer
Rock Sciences Section
Civil Reserach Department

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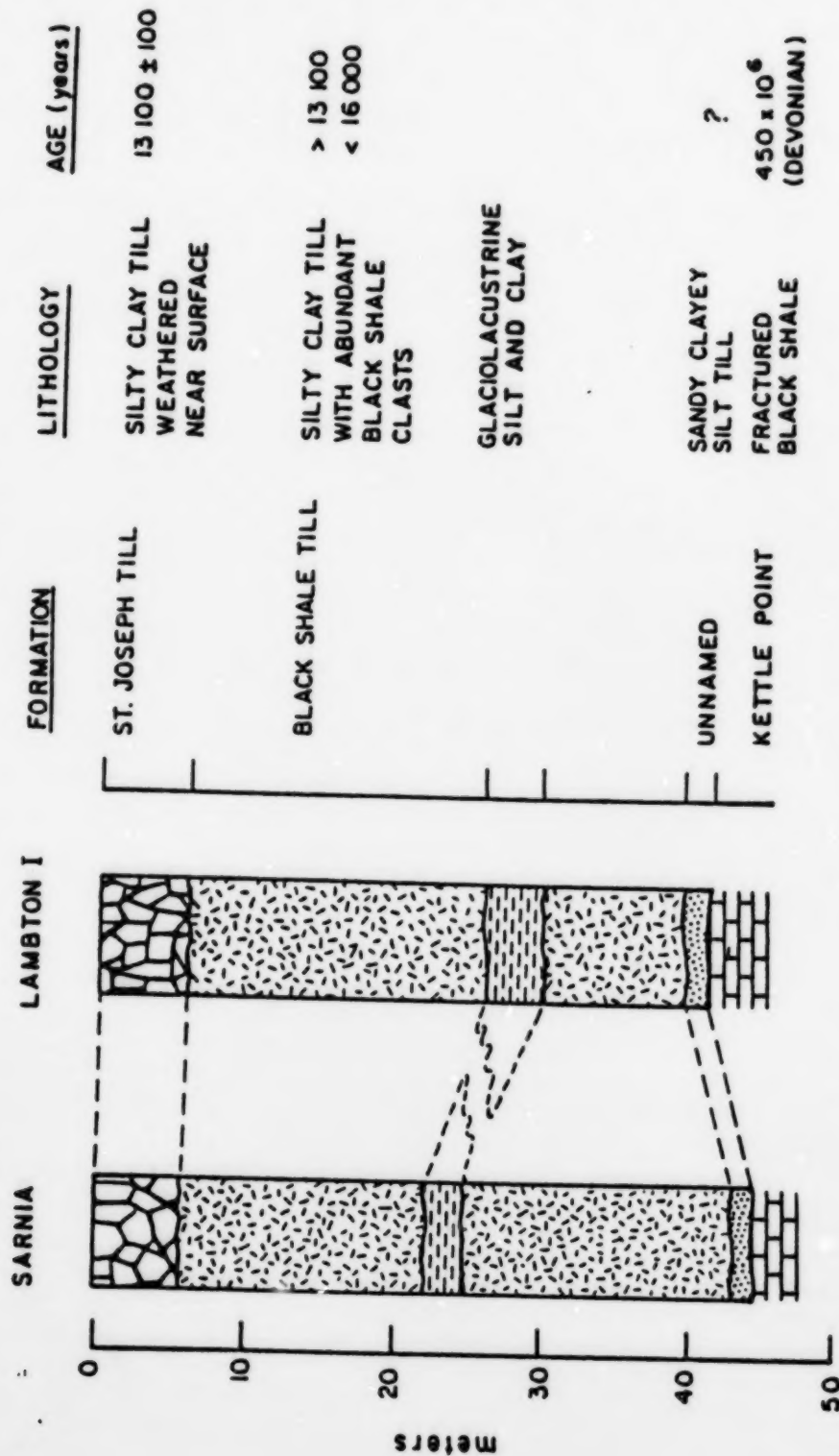


Figure 1. Stratigraphic Cross-Section of the Sarnia and Lambton Sites (from D'Astous et al., 1987)

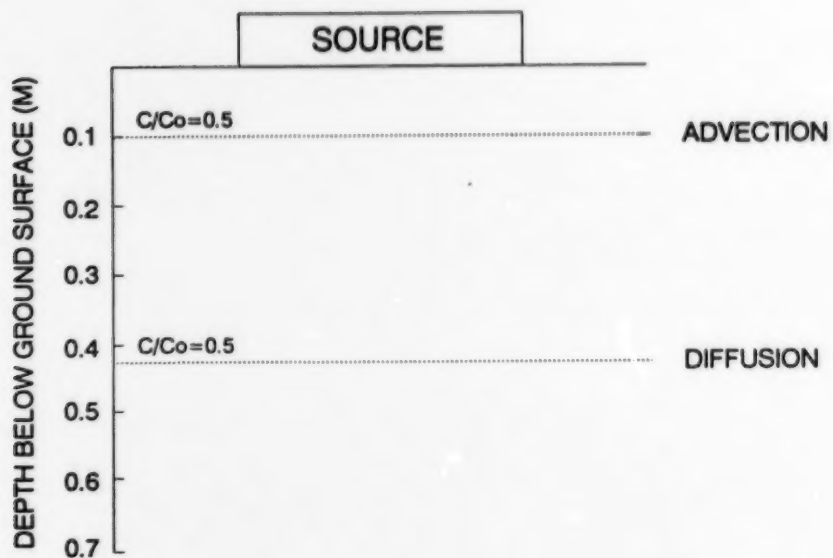


Figure 2. Effect of Diffusion in a Low Velocity Media
(average linear groundwater velocity = 1 cm/a;
effective diffusion coefficient = $6 \times 10^{-6} \text{ cm}^2/\text{s}$;
no sorption i.e. $K_d = 0$)

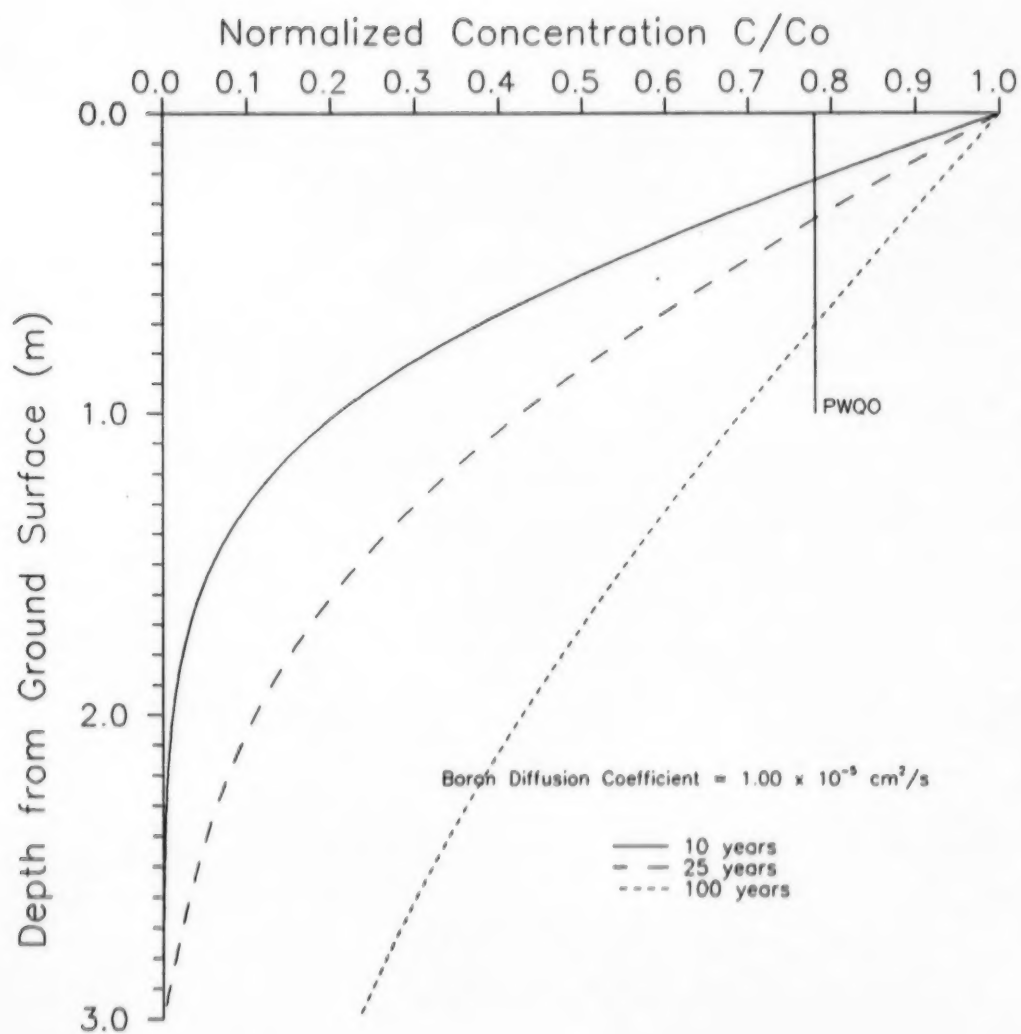


Figure 3. Calculated Boron Concentration Profile
Generated by Molecular Diffusion

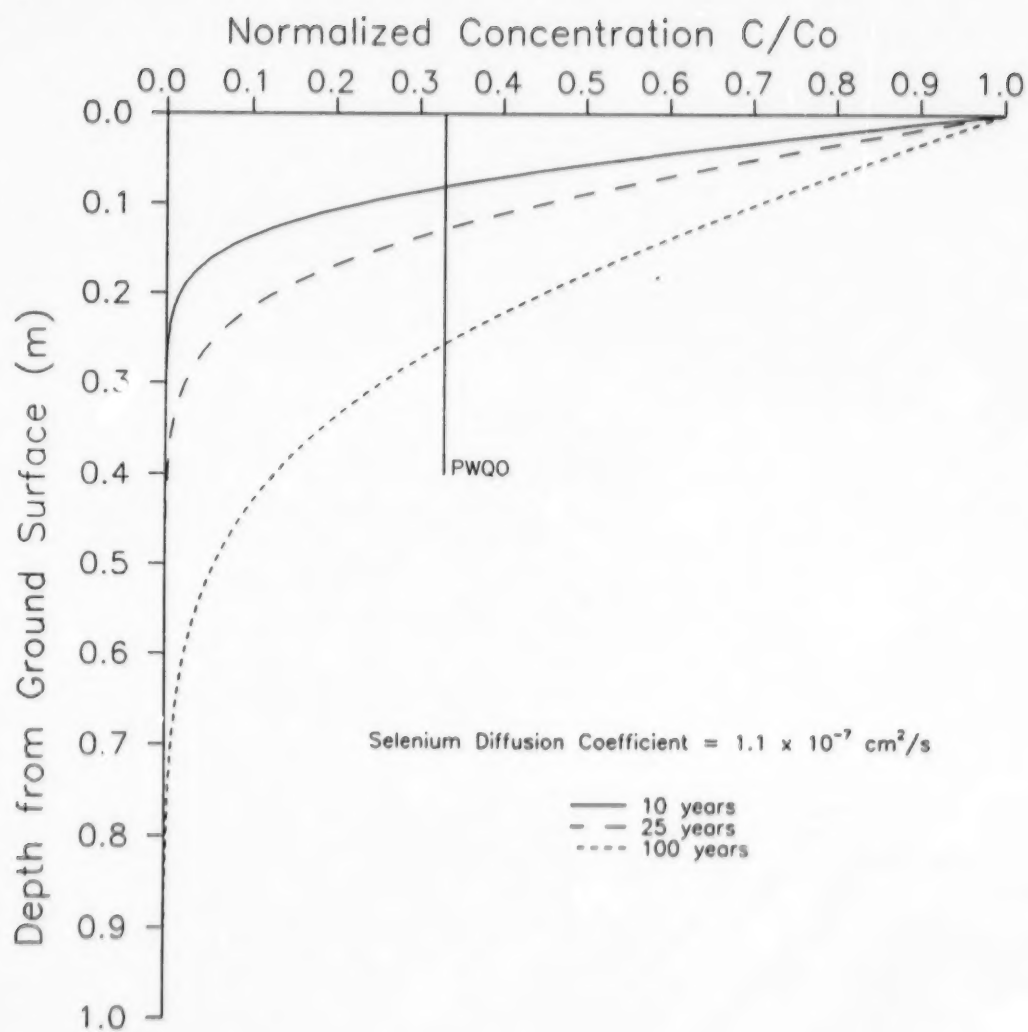


Figure 4. Calculated Selenium Concentration Profile
Generated by Molecular Diffusion

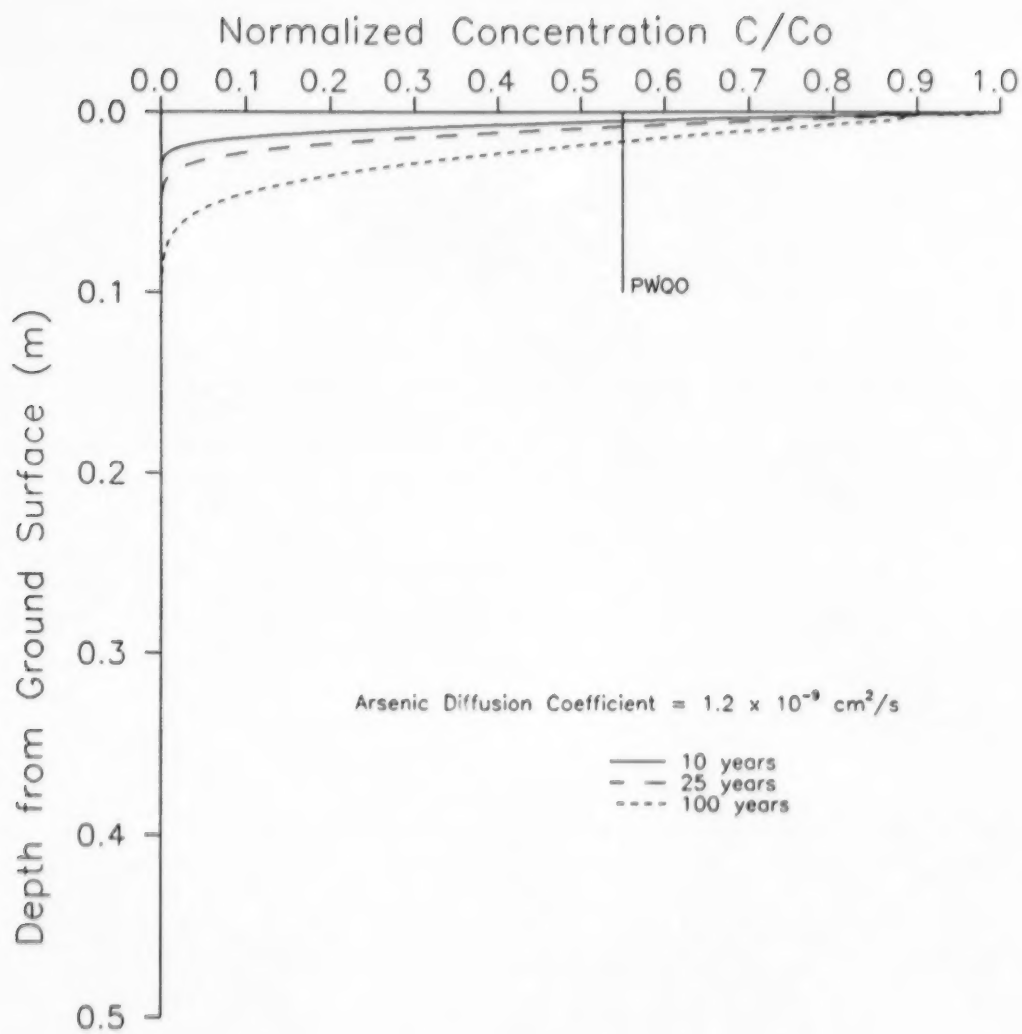


Figure 5. Calculated Arsenic Concentration Profile
Generated by Molecular Diffusion

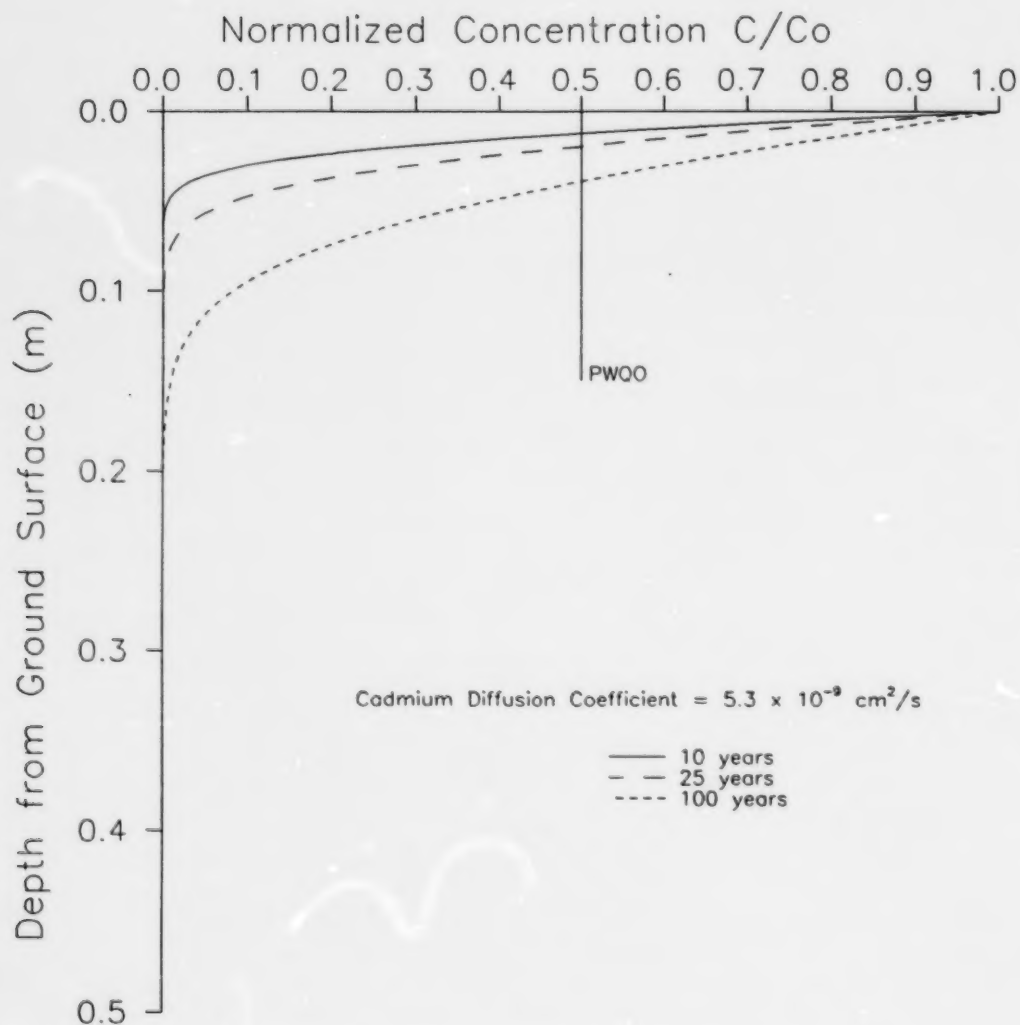


Figure 6. Calculated Cadmium Concentration Profile
Generated by Molecular Diffusion

TABLE 1**Estimated Elemental Composition of Lambton FGD Material**

Parameter	Post-Precipitator Ash ¹ (µg/g)	Lambton FGD ² Material (µg/g)	US FGD ³ Sludge (µg/g)
Ag	7.5	0.05-0.23	<8.3
As	233	1.4-7.0	0.8-53.1
B	1900	11.4-57	42-530
Cd	4.3	0.03-0.13	0.06-25
Cl	790	4.7-24	<150-8970
Cr	216	1.3-6.5	1.6-180
Cu	142	0.85-4.3	6-340
Fe	69900	419-2097	130-13800
Hg	0.5	0.003-0.015	0.005-6
F	449	2.7-13.5	266-1017
Mn	254	1.5-7.6	37.3-312
Ni	193	1.2-5.8	<5-145
Pb	158	0.95-4.74	0.25-290
Se	261	1.6-7.8	<2-162
U	7.8	0.05-0.24	-
Zn	375	2.3-11.3	7.7-612
¹ Curtis (1977) ² Estimated composition based on 0.6% to 3% fly ash content ³ Rai et al. (1987)			

TABLE 2**Estimated Composition of FGD Porewater**

Parameter	FGD Liquid Phase ¹ (mg/L)	MOE Water Quality Objectives (mg/L)	PDWO/G ² (mg/L)
Ag	0.004-0.018	0.0001	0.05
As	0.02-0.09	0.1	0.05
B	0.7-6.4	-	5.0
Cd	0.0001-0.01	0.0002	0.005
Cl	941-9901		250
Cr	0.03-0.1	0.1	0.05
Cu	0.04-0.18	0.005	1.0
Fe	0.02	0.3	0.3
Hg	0.0004-0.001	0.0002	0.001
F	0.2-0.3	-	2.4
Mn	0.05-0.2	-	0.05
Ni	0.01-0.07	0.025	-
Pb	0.007-0.03	0.025	0.05
Se	0.1-0.4	0.1	0.01
U	0.004-0.02	-	0.02
Zn	0.1-0.4	0.03	5.0
pH	4.5-6.0	6.5-8.5	
¹ Golomb (1990)			
² Ontario Provincial Drinking Water Objectives and Guidelines			

TABLE 3**Summary of the Groundwater Chemistry at Lambton TGS**

Parameter	Concentration (mg/L)	
	Weathered Zone	Unweathered Zone
Ca	260-600	118-170
Mg	130-1910	42-190
Sr	3.5-13.3	4.8-6.6
Na	73-490	189-355
K	5.0-19.5	4.0-7.6
HCO ₃ ⁻	200-1017	100-173
SO ₄ ²⁻	1113-8310	54-2227
Cl	12-366	37-809
Ag	<0.0005-0.006	<0.0005
As	0.001-0.002	0.001-0.005
Cu	0.002-0.02	0.002-0.013
Mn	0.0003-0.0017	0.00007-0.0005
Pb	<0.002-0.005	<0.002-.07
Zn	<0.002-1.6	<0.02

TABLE 4
Effective Diffusion Coefficients

	Diffusion Coefficient (cm ² /s)
Cl ⁻	6×10^{-6} (1) 6.5×10^{-6} (2) $4-6 \times 10^{-6}$ (3) $5.8-8 \times 10^{-6}$ (4)
³ H ⁺	6.2×10^{-6} (5) $6-8.5 \times 10^{-6}$ (4)
B ⁺	1×10^{-6} (6)
As	$1.2-5.6 \times 10^{-9}$ (5)
Se	1.1×10^{-7} (5)
Cd	5.3×10^{-9} (5)
* measured values, all others calculated using Equation (5.3) (1) Rowe et al., 1988 (2) Quigley, 1989 (3) Johnson et al., 1989 (4) Desaulniers, 1984 (5) Johnston, 1980 (6) Rai et al., 1989	